2013

Public Land and Its Management: Why the Research Is Not Enough

Corinne Calhoun
Pomona College

Public Land and Its Management: Why the Research Is Not Enough

Corinne Calhoun

In partial fulfillment of a Bachelor of Arts Degree in Environmental Analysis,

2012-13 academic year, Pomona College, Claremont, California

Reader:

Dr. Frances Hanzawa

TABLE OF CONTENTS:

Introduction:

> "Many studies have demonstrated the negative effects of hemiparasites on host performance, such as decreased host biomass and reproductive capabilities. Much less work, however, has examined the role of hemiparasites in shaping plant community structure, though it has been proposed that the effect of a hemiparasite can vary based on a number of factors, including nutrient availability. Using *Castilleja miniata* as a test species, I tested for an interactive effect of hemiparasite removal and nitrogen enrichment on plant community structure. I found no evidence of an interactive effect altering species richness, total plant cover, diversity, or evenness. Nitrogen availability and *C. miniata* presence or absence may have an additive effect on nitrogen-fixer productivity. Nitrogen-fixers grew more in plots under ambient soil conditions where *C. miniata* was removed. The importance of nitrogen-fixers in an ecosystem warrants further research into the dynamics between nutrient cycling, hemiparasites, and nitrogen-fixer performance" (Calhoun *et al.* 2012, unpublished).

The above quote is the abstract section from the scientific article I completed during the summer of 2012 at the Rocky Mountain Biological Laboratory (RMBL) in Gothic, Colorado (See Appendix I for full article). Like any good budding scientist, I briefly summarized two and a half months of work into one paragraph containing: the motives for my study, what I did, what I found, and where this research should go in the future. What justifies my research and how does it relate to the "real world" where land is used for a myriad of things other than ecologists' playground and being pretty? It is true that "basic ecology" such as this cannot directly inform land use policies, but it can give some insight into how and to what degree organisms interact and thus rely on each other, how organism interactions are affected by abiotic factors that may or may not be manipulated by humans, and the necessary future research that has the potential to inform management strategies.

Ecological research does not, however, tend to capture or consider human-human interactions or economic and cultural values that must be factored into land management decisions. All too often, land use recommendations informed solely by wildlife needs are not practical: land managers can face major resource limitations, they must consider other potential uses of the land in question, who will be benefiting from one use over another, etc. While conservation biologists may be mostly concerned with plants and animals, land management also must concern itself with the needs of humans. In any given landscape there are a number of stakeholders, often with conflicting agendas, and it is up to a land management agency such as the Bureau of Land Management (BLM) to prioritize such interests.

The Rocky Mountain Biological Laboratory is on private land, however the surrounding area is managed by the BLM and, because of the multi-use mission of the BLM, this land is used in a number of different ways. BLM lands support scientific research, cattle grazing, outdoor recreation (skiing, mountain biking, hiking), and resource extraction. Restoration and conservation goals are also met by the removal of invasive plant species, fire management (fuel reduction, rehabilitating burned landscapes, fire fighting), and designated wilderness and wilderness study areas.

Both scientific and non-scientific factors inform decisions regarding these land use practices, though one or the other may be given more weight in any given situation. I will not attempt to detangle the intricacies of public land policy, but rather will explore the contributions of scientific research that could be used to inform management decisions. In the following pages I will highlight 1) the ways in which ecological research, both basic and applied, can illuminate the effects of two land use practices (grazing and energy-related resource extraction), 2) in what ways these studies fall short of providing all the necessary information to inform management decisions, and 3) what other factors land managers must consider, using Colorado as a model for examining Federal land management practices.

The Bureau of Land Management:

The Bureau of Land Management (BLM) is the nation's largest manager of public land, whose goals are to "achieve a balance that serves the American public and sustains the health of the land" (BLM 2008). The BLM, established in 1946, manages more land than any other Federal organization of the United States and is, by all measures, a western entity. Of the 256 million surface acres and 700 million acres of sub-surface mineral estate managed by the BLM (Skillen 2009, p. 1), only 1 million surface and 40 million acres of sub-surface mineral estate are located east of Montana, Wyoming, Colorado, and New Mexico (Fig. 1). In contrast, the Federal government manages much of each western state. In Colorado, for example, about 36 percent is federally managed (U.S. Census of Population 1997 as seen in Bryner 1998, p. 33).

Unlike the National Park Service, which is more concerned with wildlife protection, the BLM has a mission of "multi-use" management. It is charged with providing recreation, resource extraction, and grazing lands for the American people while also restoring and maintaining land for sustainable use. This developed over time but was initiated with the Taylor Grazing Act of 1934, which charged the interior secretary to both "stabilize the western livestock industry *and* improve rangelands" (Skillen 2009, p. 7). This mandate lead to the creation of the Division of Grazing (1934-1939) within the General Land Office and the U.S. Grazing Service (1939-1946); the General Land Office and U.S. Grazing service eventually merged to form the Bureau of Land Management in 1946 (Skillen 2009, p. 7).

Since its creation the BLM has struggled with its mission as the conflicts between goals became apparent. In 1976, the Federal Land Policy and Management Act (FLPMA) attempted to bring together the BLM's various charges by defining multi-use management as, "the management of the public lands and their various resource values so that they are utilized in the combination that will best meet the present and future needs of the American people." As such, BLM lands continue to support a wide range of uses geared toward both conservation and human use, including scientific research, grazing, outdoor recreation, and resource extraction.

With a budget of more than $1 billion, the BLM is one of the few federal agencies that provide more revenue to the United States than it is given to spend. By some accounts, this makes the BLM a highly successful and valuable organization, however a strictly monetary figure does not capture the costs and benefits to each interest group. Ecological studies investigating the effects of a given land practice often illuminate the ways in which the BLM's restoration or conservation goals are not being met. Such studies can be valuable to land managers by showing the consequences of previous land use decisions. Further, more focused research, can then answer specific questions land managers have to better inform future decisions. Below, I give brief reviews of the scientific information available that is associated with two of the BLM's goals: using public lands to give grazing security to the livestock industry and resource extraction to provide revenue to both the state and Federal governments. The following sections are not meant to provide an exhaustive review of the available research. I instead focus on specific studies that serve as examples of the benefits of and the gaps found in the current knowledge base as it pertains to land management needs. I then discuss how new research can be re-focused to better serve land managers and what other factors the BLM must consider in land use planning.

Ranching and the BLM:

As a scientist working on BLM land I was concerned for my study when half way through the summer the area was opened to free range grazing. Many of the stakes and string marking my study plots were ripped out and on more than one occasion I found a fresh cow patty in the middle of my nitrogen manipulations. This is not a unique situation to be in as about 157 million of BLM-managed acres support livestock grazing (BLM 2012). On many of these acres, grazing practices can be destructive and are in direct contradiction to conservation goals. Scientific research can be helpful to land managers by illuminating in what cases and under what circumstances grazing could compromise restoration or conservation goals and in what ways it may contribute to restoration efforts. In order for this information to be useful, however, land managers are in need of site and ecosystem-specific studies.

A number of sources, reviewed below, have linked grazing pressure to changes in both plant and animal community structure as well as altered nutrient dynamics and carbon sequestration by soil. Large herbivore grazing, specifically livestock, has the potential to greatly change an ecosystem. Because of this impact on habitat structure, some suggest using livestock for conservation or restoration purposes (Carlson and Cringan 1975; BLM 1996; Collins et al. 1998; Goguen and Mathews 2001). However, as will be revealed, grazing by domestic ungulates can have greatly different effects based on ecosystem type. With such important implications of grazing regimes, managers should be aware of how defoliation, fertilization, and trampling by cattle, the main mechanisms behind environmental alteration by large grazers (Sørensen et al. 2009), will impact the land in question.

Unfortunately, as I will discuss in the following sections, basic research does not always provide enough information or the right information to fully inform such management decisions. Rather, applied study of the given ecosystem is more useful. I will discuss this theme by providing examples in the literature concerning the effect of grazing on the vegetation structure, animal community, and nutrient dynamics of impacted land. I will then discuss what further research is necessary as well as the non-scientific factors informing management decisions.

Effect of Grazing on Plant Community Composition:

There is no shortage of studies that investigate the effects of large herbivore grazing on community structure and biodiversity. The measured change in biodiversity (of plants or animals) of a given activity is often used as an indication of how sensitive an environment is to changing land use practices. In Oregon, for example, Kauffman et al. (1983) measured, among other variables, vegetative biodiversity in non-grazed and late-season grazed riparian vegetative communities to determine the effect of grazing in this habitat. This study found both increases and decreases in frequency of some plant species with grazing pressure, though there was no significant difference in total plant species diversity or the number of species present (Kauffman et al. 1983). The ability for grazing to change community composition, however, can vary in different ecosystems (Milchunas and Lauenroth 1993). In shortgrass steppe plant communities in Colorado, for example, overall plant diversity decreased with grazing and was also linked to topography: cattle grazed less on ridge tops (Milchunas et al. 1989). Ideally, then, management decisions would be based on site-specific or ecosystem specific studies that correspond to the land in question.

By looking more deeply, one will find that individual plant species or groups can show varying reactions to grazing pressure as well. Such varied responses contribute to changes in dominance and competition. *Bouteloua gracilis* is a grazing and drought tolerant native grass species (known as a C_4 species) that increased in dominance with grazing on a shortgrass steppe in Colorado (Milchunas et al. 1998). In that same system, Reeder et al. (2004) found that sites excluding cattle (exclosures) demonstrated the opposite plant types, dominated by “cool season” C_3 grasses. In addition, an earlier study showed opposite reactions of grasses and forbs to grazing in the same ecosystem type (Milchunas et al. 1989). Grasses were more abundant under the grazing regime and forbs more abundant in non-grazed sites. It is plausible, then, that grazing could be managed in such a way as to produce a particular desired landscape, or to manage certain plant groups.

In fact, because plant groups respond differently to grazing, some have suggested grazing may provide an important ecosystem function of managing invasive plant species (BLM 1996; Collins et al. 1998; Germano et al. 2001). Cattle graze on both living plant matter and mulch, which aggregates from mostly alien grasses in California and may inhibit germination and growth of native species (Evans and Young 1989 as seen in Kimball and Schiffman 2003; Foster and Gross 1998). Grazing would presumably decrease the negative effects of mulch on natives. If land managers proceeded to make management decisions with this assumption alone, however, they may be sorely disappointed.

Despite the logic, grazing in a California grassland proved to be an ineffective management tool of invasive species: a three year clipping and mulch removal manipulation resulted in decreased species richness, and relative cover of natives due to clipping and prompted inconsistent responses of natives to mulch removal; while introduced plants responded little to either treatment in terms of species richness and cover (Kimball and Schiffman 2003). In contrast, Colorado grasslands showed decreased density of introduced species in grazed sites after 45-years of controlled heavy grazing versus exclosure (Milchunas et al. 1989).

Results such as these have been attributed to the evolutionary history of species in the Colorado landscape, which evolved when wild herds of bison grazed the steppe (Milchunas et al. 1989; Milchunas et al. 1998). In contrast, landscapes west of the

Rockies lost much of their large grazers post-Pleistocene era along with the plants that had adapted to their presence (Kimball and Schiffman 2003). Introduced grass species in California are also mostly from Europe, where they adapted to sheep grazing (Kimball and Schiffman 2003). Cattle simulate bison grazing in diet preferences (though not always in frequency due to human manipulation; Laurenthal and Milchunas 1991 as seen in Milchunas et al. 1998) and may be a necessary restoration tool in historically grazed landscapes.

These studies exemplify the necessity for site or ecosystem-specific research in order for land managers to make informed decisions regarding grazing practices. Basic research regarding specific plant species' evolutionary histories can also inform management if surveys of present plant species are conducted before grazing is implemented. Knowledge of plant species' origins, generally available in the literature, can help managers avoid degrading a landscape further. Animal communities are also altered by grazing, however, and must be considered by land managers as well, especially if the site in question supports threatened species.

<u>Effect of Grazing on Community Composition of Select Animals:</u>

Changes in plant community composition caused by grazing pressure may alter the accompanying animal communities as well. Many plants, both wild and domestic crops, for example, are insect pollinated and thus support and depend on these invertebrates. Because of their great importance in a plant's ability to successfully reproduce, insect pollinators should be considered in management decisions concerning livestock grazing. A study by Kearns and Oliveras (2009) investigated insect abundance with respect to proximity to urbanization in Boulder County, CO. Sites were either not grazed, or grazed for one, three, or 20 plus years. The study found that bee abundance decreased with increased levels of grazing and sites that had been grazed for one year had the highest mean abundance of ground nesting bee (Kearns and Oliveras 2009).

Similarly, other aboveground macroarthropods have been shown to increase in abundance with light grazing, but decrease with moderate and heavy grazing in central Colorado (Lavigne et al. 1972 as seen in Milchunas et al. 1998). Duration and intensity of grazing, then, play an important role in determining community composition changes in response to grazing. Managers have the simple choice to allow grazing or not, but they also can determine with what intensity a site is grazed. As seen above, managers should put great thought into the density of cattle and duration of grazing.

The mere presence of cattle, however, may affect sensitive or threatened species. A major conservation concern in southwestern Colorado and southeastern Utah is the potential negative effects of domestic grazing on the Greater and Gunnison sage grouse populations, which are in decline. Livestock have been shown to have both positive and negative effects on sage grouse populations (Beck and Mitchell 2000). While there is little direct evidence that associates grazing practices with sage grouse population declines, Beck and Mitchell's (2000) review found that trampling by cattle can damage sagebrush, particularly smaller plants, and grazing can alter community composition in such a way as to decrease suitable nesting habitat for the birds. In addition, Klebenow (1985) found that long-term population declines in Nevada were related to low reproduction. More research is needed to tease apart the direct and indirect effects of grazing practices on sage grouse populations if scientific research is going to provide guidance to land managers. Specifically, studies should determine how prime sage grouse habitat serves the species and how grazing may or may not alter that functional ability.

Effect of Grazing on Soil Carbon Sequestration and Nutrient Dynamics:

In light of global climate change, it is important to note the growing body of research investigating the carbon (C) sequestration potential of various ecosystems. As such, land managers should consider the sequestration characteristics of land under different management regimes. Unfortunately, the current body of research is inconclusive as to what grazing regimes may increase carbon sequestration and by what mechanism.

Defoliation of grazing-tolerant species can increase photosynthesis, which removes carbon dioxide (CO_2) from the atmosphere (Painter and Detling 1981). Photosynthesis, however, is also influenced by the amount of nitrogen (N) available to the plant, which depends on the rate at which organic N is converted by microbes into usable inorganic forms (NH_4^-N and NO_3^-N). Reeder et al. (2004) found a greater mass of NH_4^-N in both lightly grazed and heavily grazed treatments compared to paired exclosures while NO_3^-N mass was lower in lightly grazed sites and higher in heavily grazed sites compared to exclosures. They also found greater ratios of microbially respired CO_2 to net mineralized N in both grazing treatments compared to nongrazed treatments (Reeder et al. 2004). However, in this system the proportion of soil organic N that is converted to inorganic N (the process of mineralization) is lower in grazed versus nongrazed treatments (Reeder et al. 2004). They conclude that grazing alters soil organic matter but also stimulates precipitation of inorganic carbonates, therefore soil inorganic C in addition to organic C must also be considered when determining the effects of grazing on carbon sequestration.

As I mentioned above, different ecosystems show varying responses to grazing. Milchunas and Lauenroth (1993) compiled a quantitative review of global data, which revealed decreases <u>and</u> increases in soil N and C pools with grazing. Piñeiro et al. (2010) also found a wide range of responses. This is, in part, because of climatic factors that also work with grazing pressure to alter ecosystem response, including soil moisture (Holland and Detling 1990), temperature, and precipitation (Piñeiro et al. 2010). Furthermore, N and C concentrations showed opposite responses regardless of the degree of change in species composition (Milchunas and Lauenroth 1993). Because of this, the authors caution against using only measures of species composition to assess the impact of grazing and a rangeland's ability to sustain productivity.

In making management decisions in the Colorado shortgrass steppe, the BLM might look to LeCain et al. (2002). While this study found seasonal differences between grazed and non-grazed sites in C exchange rates, there was no difference when data were averaged over each growing season. This was attributed to the lack of change in photosynthetic surface area with grazing in the shortgrass steppe ecosystem. In contrast, Bremer et al. (1998) showed a decrease in canopy photosynthesis and productivity in mid- to tall-grasslands. As with plant community, it would seem ecosystem type is incredibly important in determining the effect of grazing on C sequestration and nutrient dynamics. As is the available research does not give a clear picture to land managers. If the information were desired, the BLM could fund research for the ecosystem in question on a smaller site to understand grazing effects on soil C and N before making a decision that would apply to the greater area.

Improving the Research:

These studies are not meant to be a complete review of grazing literature, but merely serve as examples of how ecology can inform management decisions and where information is lacking. As was revealed above, site-specific studies would be most helpful to land managers. However, given potential funding constraints and interest of ecologists this is not always possible. For management decisions regarding shortgrass steppe habitats in Colorado, the BLM can benefit greatly from studies conducted at the USDA Agricultural Research Service Central Plains Experimental Range. In my literature review, however, I found it much more difficult to find studies conducted in montane and alpine habitats let alone mountainous ecosystems in Colorado specifically. For decisions regarding Gunnison County, then, land managers may be forced to extrapolate from studies conducted in montane ecosystems elsewhere (such as Wyoming).

Site or ecosystem-specific studies can be important for a number of reasons; one highlighted here is the varied evolutionary history of land in the United States. Evolutionary history of grazing was a major factor in determining ecosystem response to grazing. As such, the BLM cannot implement similar grazing regimes west of the Rockies as it does in Colorado and hope to attain similar restoration goals. Restoration is only one of many goals of the BLM, however, the others driven by factors other than ecological implications.

Non-Scientific Factors Influencing Management of Livestock:

As in much of the American West, ranching is a large part of the Gunnison County economy and grazing interests have historically dominated BLM policies. With the creation of the Division of Grazing in 1934, much of the power to make land management decisions was given to ranchers in the form of advisory boards made up of grazing permit holders (Skillen 2009, p. 7-8). The director of the Division of Grazing at that time, Farrington Carpenter, entrusted these advisories with day-to-day management decisions and in so doing handed over power to the beneficiaries of specific public lands uses. The Taylor Grazing Act, passed at the same time, also gave the tasks of stabilizing livestock industry <u>and</u> restoring rangelands to the agency that is now the BLM. Given the power of ranching interests in BLM politics and the need to give ranchers grazing security in order to stabilize the industry, restoration goals were often neglected in favor of opening more public land to grazing leases.

Data contributed by both basic and applied studies indicate that, in landscapes adapted to large grazers, restoration and livestock industry stability need not conflict as heavily as they might in, for example, California. In Colorado ecosystems, grazing may prove to be an important restoration tool, however it must be managed with appropriate duration and levels of intensity. These qualifications, unfortunately, are often more influenced by economic need rather than restoration needs.

Conflicts between goals such as these arise when one interest group is given more power within BLM politics. In the past the BLM has had legal power but little political power since, at the time of its creation, Congress still supported self-regulation by grazing permittees. As the BLM grew and environmental concerns came into the picture with the National Environmental Policy Act (NEPA) in the 1980s, it became a debate over decentralized local control versus centralized management, which was generally supported by BLM leaders and conservation and environmental groups (Skillen 2009, p. 10, 12).

Fast-forward fifty plus years: the BLM may be a bit more autonomous, however National priorities that inform BLM goals and actions have noticeably shifted. While ranching is still a major interest group, Federal policies dictate that the BLM places a stronger focus on resource extraction. Giving priority to resource extraction over grazing creates friction between ranchers and miners, tension that must be navigated by land managers of the BLM. In addition, resource extraction interests may prove to be even more in conflict with conservation and restoration goals of the

BLM. As I did for grazing interests, in the following section I indicate how scientific research can and cannot inform management decisions regarding extraction of natural gas as well as the non-scientific factors that influence BLM decisions, often at the expense of public health and the environment.

Energy and the BLM:

It is no secret that mining activities contaminate ecosystems and affect the surrounding environment. Resource extraction can affect the landscape in a number of ways, including surface disruption (Cypher et al. 2000) and contamination of water sources (Dammel et al. 2011). For example, lead-zinc mining has been linked to changing fish community compositions in Missouri streams (Allert et al. 2009). Mining also tends to produce undesired byproducts that are often toxic and must be disposed of. Disposal of byproducts is one mechanism by which the environment can become contaminated. Open disposal of oil shale byproducts, for example, is problematic due to the possibility of arsenic leaching into groundwater (Datangel and Goldfarb 2011).

Despite the negative side effects, these and similar mining efforts continue to exist because they can provide important economic boosts, both locally and nationally, and the raw resources extracted are ultimately the building blocks of our society. In addition, overall extraction of resources used for energy production (coal, oil, and natural gas) is becoming increasingly more prominent in the U.S. as we focus on becoming an energy-independent nation. Given the seeming permanence of such practices, research should focus on minimizing the negative effects. It can be argued that some mining practices are less destructive than others. Coal, for example, is generally considered the "dirtiest" energy source, both in production and burning (Bryner 1998, p. 200). Natural gas burns cleaner but the ecological and health effects of its extraction are not well understood.

A full review of the effects of various techniques for extraction of each resource is not feasible in this space. Because of recent increases (and predicted increases) in natural gas extraction, below I highlight some of the literature available on the ecological and health effects associated with extraction and what still needs to be determined.

Unconventional Sources of Natural Gas:

Domestic extraction of natural gas has increased significantly, especially in Colorado, Texas, and Wyoming, since 1999 (Weber 2012). New estimates by the Energy Information Administration (EIA) indicate the U.S. has enough natural gas resources to last 110 years (U.S. House of Representatives Committee on Energy and Commerce 2011). Of the unconventional sources of natural gas (coal-bed methane, tight sands, and shale gas) that we can now access, shale is growing most in use (Gregory et al. 2011). Shale gas is predicted to contribute up to 45 percent of domestically produced natural gas (total ~26 trillion cubic feet per year, up from ~24 trillion cubic ft) in the year 2035 (EPA 2011).

Producers use hydraulic fracturing, or 'fracking,' in order to access natural gas trapped in shale deposits. Fracking is a sixty-year-old technique that has increased in use due to decreased costs of production. More recently, fracking has been used in conjunction with horizontal drilling in order to reach previously inaccessible natural gas embedded in shale deposits. Horizontal drilling means fewer wells are needed to extract gas from the same formation. A single horizontal well can replace three or four vertical wells (Arthur et al 2008), which may decrease the negative ecological effects incurred during the pad-site construction, drilling, and well development processes (Gregory et al 2011).

Fracking, however, has a different suite of risks that may have negative effects on both public health and the environment. Hydraulic fracturing entails fracturing shale with chemical infused water at high pressures and sand, which keep fissures open to then provide an avenue for the gas to reach and exit the well (Fig. 2). These 'fracking fluids' can include "commonly used household or personal care items" that are not considered hazardous (Stevens 2012) as well as other compounds that are known toxic chemicals and carcinogens (Forrest 2011). The use of these chemicals raises concerns about the potential health effects in the event of water contamination. And, indeed, much of the literature shows a strong public health concern. The EPA is currently conducting a study, which will be available in 2014, investigating the effects of fracking on ground and drinking water. This study will examine each step of the process as it relates to water (Fig. 3) and determine where the risks of contamination lay. Below, I indicate what information is currently available regarding the effects of fracking on water sources and air quality, as well as the effects observed through exposure to fracking fluid during disposal and spills.

Effect of Hydraulic Fracturing on Water Quality:

Drilling for natural gas can take place anywhere between 3,000 and 15,000 feet below the surface, which means shallow aquifers accessed by drinking wells (usually at a depth of less than 1,000 feet) may be at risk of contamination (Forrest 2011). There have also been reports of flammable drinking water, due to elevated hydrocarbon gas concentrations, and human and animal health problems following direct exposure to waste water or contaminated freshwater (Bamberger and Oswald 2012).

Drinking Water: In areas of active fracking in northeastern Pennsylvania, Osborn et al. (2011) discovered evidence of methane contamination in shallow drinking wells. Methane concentrations were on average seventeen times higher in wells located within 1 kilometer (km) of active extraction areas than in wells in nonactive areas (Osborn et al. 2011). While the mechanism behind the higher levels remains unexplained and hotly debated (Schon 2011; Davies 2011; Jackson et al. 2011; Saba and Orzechowski 2011; Osborn et al 2011b) the evidence is consistent with a deep thermogenic source of methane in the active sites. This is in contrast to the believed microbial source of (lower) methane concentrations from wells in nonactive sites (Osborn et al. 2011).

One point that has been used to speak against the validity of this study is the lack of pre-fracking data regarding the baseline conditions before production. While these responses to Osborn et al. (2011) are likely biased towards the energy industry, they still highlight a major flaw in current scientific research investigating the effects of fracking. If land managers are going to rely on similar studies in the future, baseline conditions need to be established to show causality, not just correlation as Osborn et al. (2011) was limited to.

Similarly, observational studies that document adverse health conditions of those living in fracking areas should be accompanied with monitoring of water before and during production. Humans potentially exposed to toxins related to natural gas extraction through drinking, cooking, and bathing water often describe similar respiratory and gastrointestinal symptoms (burning in the nose and throat, vomiting, diarrhea, etc.) as well as rashes and nosebleeds (Bamberger and Oswald 2012). These symptoms, however, cannot be causally linked to fracking without pre-fracking data.

Streams, Lakes, and Ponds: A recent study (Bamberger and Oswald 2012) documented cases (in Colorado, Pennsylvania, Texas, and other states) of human and animal health problems potentially due to natural gas extraction in the area. Most of the problems are believed to be due to exposure to contaminated springs, ponds, and creeks (Bamberger and Oswald 2012). Interviews with farmers revealed that they are concerned and have noticed negative effects of fracking on cattle, goats, llamas, poultry, cats, dogs, and other animals (Bamberger and Oswald 2012). In one case, a farmer had separated his heard into two pastures. Of the 60 cattle pastured in a field with access to a fracking-contaminated source of water, 21 died and the survivors experienced elevated rates of stillbirths and birth defects in calves (Bamberger and Oswald 2012). Though not conducted as a formal experiment, this is the closest thing to a controlled study showing the impacts of stream contamination by fracking fluid on animals that I found in the literature. It is important to have established controls in any study, however ethical issues often negate this possibility. In the absence of controlled experiments, then, it becomes even more imperative that more observational studies are accompanied with information regarding conditions before contamination.

Effect of Hydraulic Fracturing on Air Quality:

There is not just concern about water contamination, but air contamination as well. Natural gas production has been hypothesized to contribute to elevated levels of ground-level ozone in northeastern Utah (Weinhold 2012). In addition, chemicals released during the hydraulic fracturing process have been linked to asthma, respiratory problems, and cancer (Vergano 2012). The EPA recently mandated that gas venting from newly drilled wells must be collected or burned before actual production begins, a rule that is expected to cut 95 percent of the smog-related chemicals released during hydraulic fracturing but will be phased-in by 2015 (Vergano 2012).

Meanwhile, natural gas extraction may still be harming our health. Bamberger and Oswald (2012) documented cases of both animal and human respiratory problems associated with natural gas extraction sites. Similarly, McKenzie et al. (2012) conducted a human health risk assessment of air emissions from development of unconventional natural gas resources in Colorado. This study found that the risk of negative health effects is greater in residence living less than a half mile from unconventional natural gas development as apposed to those living more than a half mile away (McKenzie et al. 2012). In addition, benzene was considered the main contributor to the risk of developing cancer, which was 10 and 6 in a million for close and far residents, respectively (McKenzie et al. 2012). Results such as these indicate to land managers that fracking should not be located within a certain distance of residential areas.

Effect of Other Forms of Exposure:

Other forms of exposure occur during some disposal processes. In a risk assessment study of the Marcellus Shale (Fig. 4), Rozell and Reaven (2012) determined the most likely mode of water contamination is during wastewater disposal. It was "very likely" that at least 200m^3 of hydraulic fluid would be released from an individual well (Rozell and Reaven 2012). That fluid then needs to be disposed of. This is achieved in a number of ways and to varying degrees depending on the state.

In the majority of the U.S., water produced during gas extraction is generally injected deep underground (Clark and Veil 2009 as seen in Gregory et al. 2011), however it can also be processed and reused. In Colorado no waste is sent to water treatment plants, 60 percent is injected into deep waste wells, 20 percent is allowed to evaporate from lined pits, and 20 percent is discharged as usable surface water (BLM 2011a).

I was unable to find information in the scientific literature regarding the ecological or health impacts of these disposal methods specific to natural gas extraction. There is no lack of news reports, however, implicating fracking in mass bird deaths and animal harm. In Ohio, for example, fracking fluid can be legally applied to roads to control dust and ice and observers report cats and dogs being attracted to the high salt content of wastewater and subsequently becoming ill after exposure (Debatin 2012).

Direct exposure to fracking fluid during spills has also been connected with the death of cattle and fish (Bamberger and Oswald 2012). In addition, the same study documents the case of goats suffering from reproductive problems over the course of the two years following a fluid tank spill that leaked hundreds of barrels of fracking fluid into the pasture. In Colorado, John Brokerick, Division of Wildlife senior terrestrial biologist, points to waterfowl deaths as the result of exposure to reserve and production pits, which likely contain "oily hydrocarbons and toxic substances such as benzene" (CCLC, "Wildlife Impacts").

Improving the Research

Evidenced by the minimal review above, relatively few peer-reviewed articles can be found regarding the effects of hydraulic fracturing. Part of the lack in systematic studies is due to nondisclosure agreements regarding the content of fracking fluids (Bamberger and Oswald 2012). In addition, field studies are often flawed in that they begin post spill (Teal and Howarth 1984) or development, and therefore lack appropriate controls (Davies 2011). Bamberger and Oswald (2012) similarly noted that their study could not be systematically controlled, "as one variable could not be changed while holding all others constant." Many of the available studies are, instead, observational and do not have baseline data to compare results to.

Clearly lacking from the research pool are thorough pre- and post-development long-term studies that could then be used to inform future projects. Osborn et al. (2011) suggested more long-term monitoring of ground water before, throughout, and after extraction to both establish baseline data and identify the mechanisms behind groundwater contamination. Monitoring should also include data on both chemical and biological factors (Orupold et al. 2012) in order to see the full scope of environmental impacts.

Recording baseline data, especially in long-term studies, yields incredibly valuable information regarding conditions before disturbance. As I mentioned above, this is information that is lacking in far too many resource-extracting areas, likely due to funding constraints. If the BLM made funds available for basic research on key sites that may be developed in the future, it would be in researchers' best interest to conduct the necessary basic research. Currently, without such basic research, applied research post-production is little more than a description of correlations and may be less helpful for land managers when, for example, they must approve reclamation efforts.

Currently, the BLM has a number of set protocols to ensure reclamation upon conventional oil drilling abandonment: baseline community data are collected before approving a permit to drill and construction begins; inspections are performed at least every three years, sites prioritized by "potential health and safety issues, environmental concerns, potential conflicts with other resources, and compliance history" (BLM 2009). Lease-holders are also encouraged to make regular internal inspections and practice reclamation from the beginning to ease final reclamation. I was, however, unable to find similar protocols indicated on the BLM website regarding hydraulic fracturing for natural gas. In fact, the BLM was taken to court under accusations of not conducting an analysis of air pollution risks before authorizing drilling in three projects near the Colorado Roan Plateau in 2011 (Investigative Newswire 2012). The BLM had, instead, relied on an environmental impact statement from 2006 that did not mention any of these drilling locations (Investigative Newswire 2012).

Non-Scientific Factors Influencing Management of Natural Gas Extraction:

Despite the ecological costs of production and limited site-specific thorough studies, resource extraction practices are a growing part of the national economy. The production of coal, crude oil, and natural gas has increased significantly in the past sixty years: from 1950-1994 alone, crude oil extracted on all Federal land (including acquired military, Outer Continental shelf and public lands) has increased from 5.4 to 22 percent, natural gas from 2.4 percent to 36.2 percent, and coal from 1.4 to 31 percent of the national total (U.S. Department of Energy 1995 as seen in Bryner 1998, p. 179). The BLM, specifically, manages extraction of oil and gas, coal, and other non-energy minerals. In fiscal year 2010, management of the National System of Public Lands produced $40 billion worth of energy and non-energy minerals, which boosted the U.S. economy by an estimated $103 billion (BLM 2011b).

The 2011 BLM factsheet, “The BLM: A Sound Investment for America,” highlights economic gain from its land use regimes, broken down by state. Resource extraction from BLM land and subsurface holdings in Colorado includes oil and gas, coal, and non-energy non-hardrock minerals, from which it generated an estimated $2,930.9, $782.7, and $17.6 million respectively to the state economy in fiscal year 2010 (BLM 2011b). However, in Colorado and other arid landscapes availability of water for use in natural gas extraction can impede or decrease feasible hydraulic fracturing practices. The BLM or other management agency may, then, need to consider other forms of revenue-building resource extraction. Despite these limitations, 95 percent of new wells in Colorado are fractured (BLM 2011a).

Feasibility of extraction of energy sources from shale gas may also depend on disposal techniques available. Above I reviewed the negative effects of some disposal methods, but disposal can also be hindered by lack of physical infrastructure able to process byproducts (Arthur et al. 2008), which then becomes a factor to consider in management decisions. In underground injection of wastewater, deep-well capacity can be a constraining factor. In the Marcellus Shale region, deep-well capacity is limited so out of necessity producers in that region depend more heavily on processing facilities, despite the fact that they were not originally designed to process fracking fluids.

In addition, on-shore oil, gas, and energy-related mineral extraction managed by the BLM contribute toward the U.S. becoming energy independent. Natural gas consumption is on the rise, in part due to its lower greenhouse gas emissions than coal combustion (Jaramillo et al. 2007) but also because it is seen as an important step in becoming secure in our domestic energy needs. It currently supplies 24 percent of the total energy demand in the U.S. (EIA 2011 as seen in Gregory et al. 2011) and in 2011 94 percent of the natural gas consumed in the U.S. was produced domestically (EIA 2012). The newly accessible shale formations make natural gas a tempting source of energy that is less vulnerable to transportation interruption due to international disputes.

With a projected increase in natural gas production of 29 percent, most of which is due to growth in shale gas production (EIA 2012), it is imperative that we approach production responsibly. This means funding and <u>using</u> scientific research that further illuminates the risks associated with hydraulic fracturing, including baseline studies of pre-production conditions to facilitate appropriate remediation post-production. Saying we need these things, however, does not make them so and furthermore, it does not address the issue of often conflicting influences and interests informing public land management.

<u>Rectifying Scientific and Non-Scientific Influences in Public Land Management:</u>

The BLM is wedged between multiple stakeholders and has seemingly contradictory goals with various informing factors. In the two previous chapters I used grazing and resource extraction as examples of land use practices whose policies are dictated by more than just ecological research. I indicated in what ways research might be expanded to provide more useful data for land managers as well as highlighted the other factors that the BLM and other management agencies must consider when allocating leases and permits. I am not the first to acknowledge the conflicts that arise from a multi-use mission, but the question remains, "How can these conflicts be resolved?"

Boundary spanning:

Others have proposed that ecological research is lacking in some areas and, thus, is not always helpful to land managers. Nelson (1999), for example, discusses how "science has not evolved to the point that it can provide decisive evidence proving the validity of one management approach over another." While the nature of scientific inquiry and presentation of results will never lead to a step-by-step guide for managers, the focus of research can stand to be expanded in order to answer more pertinent questions to land managers specifically. There are a number of reasons why this is not already always the case, including funding limitations, interest within the scientific community, and communication between researchers and land managers. To hurdle the last obstacle, some (e.g. Driscoll et al. 2012) have suggested and supported efforts of "boundary spanning," communication and interaction between researchers, decision makers, and society in order to focus research on more relevant questions that can then inform environmental planning.

The NSF-funded Long Term Ecological Research (LTER) Network is an example of successful cross-disciplinary communication with decision makers and the public. The Network's twenty-six sites focus on creating long-term data sets that investigate primary production, population studies, movement of organic and inorganic matter, and disturbance patterns across a wide range of ecosystems (LTER 2012). Driscoll et al. (2012) discuss how this thirty-year-old collaboration has greatly contributed to “science-policy integration efforts.” For example, by conducting long-term monitoring of stream water and precipitation chemistry at the Hubbard Brook Experimental Forest, the LTER Network contributed to assessing the effectiveness of the Clean Air Act of 1972 and its amendments. Importantly, this research led to the creation of Science Links, which brings together scientists from a wide range of disciplines and policy advisors to investigate policy-relevant research questions (Driscoll et al. 2012).

Problems with Boundary Spanning as a Solution:

The communication between scientists and policy-makers has improved over the past few decades, however ‘boundary spanning’ and the LTER Network are only part of the solution. While these, and similar, approaches do address the researcher-decision maker relationship and place emphasis on long-term studies, other stakeholders, such as resource-users (ranchers, mining companies, and recreationers) are noticeably absent from the analysis put forth by Driscoll et al. (2012).

In addition, because research is conducted at a limited number of specific sites, these long-term studies at LTER Network sites are less helpful in establishing baseline data for other landscapes that may be developed in the future. The latter issue may be solved by allocating more funds to land management agencies, such as the BLM, to conduct the necessary basic research. Managers are in the best position to prioritize sites for initiation of baseline studies, as they know which sites will likely see development in the future. As for the former issue, including other stakeholders in the circle of communication, a broader approach than 'boundary spanning' may be necessary.

Transdisciplinary Approach:

Land use practices are inherently human-oriented and thus decision makers must consider a number of political and socioeconomic constraints, as well as the values of the people they are representing. In order to facilitate decisions that take all these factors into account, some (e.g. Reyers et al. 2010) call for more bidirectional communication between all the pieces of the puzzle, or a "transdisciplinary" approach.

Multi- and inter-disciplinary are often used interchangeably and can be equated with the aforementioned 'boundary spanning' approach. These two concepts, defined by Lengwiler (2006) and Max-Neef (2005) (as seen in Reyers et al. 2010), are cooperation between disciplines and cooperation and <u>exchange</u> between disciplines, respectively. Transdisciplinary research takes these concepts one step further and incorporates both scientific disciplines and other "knowledge spheres" to inform and feed into each other without separation of knowledge (Reyers et al. 2010). This approach rests on a theory of interaction between different spheres of knowledge: the empirical level (traditional scientific and social research) provides information to improve the pragmatic level (technology, agriculture, forestry, tourism, etc.), which is managed by the normative level (land use planning, policy, law). Furthermore, these three are all ultimately dictated by the societal values that determine what is acceptable and desired.

Management of public lands is very much connected with these themes. Values held by the Federal government, for example, have more recently placed resource extraction above ranching interests because of the desire to become an energy independent nation. Given that the U.S. is a democracy, ideally, these values would reflect those held by the American people. Unfortunately it is not that simple. We are a diverse people with diverse priorities. To make it more complicated, these priorities are not set in stone. They are shaped and altered by where we live, changed conditions (such as gas prices and oil spills), and the way in which those events are depicted in the media (Fitzsimmons 2012, p. 71-77). Landowners and ranchers, for example, have recently been opposed to energy development resource extraction because it impedes on their surface land rights (Forbis 2010). Currently, in the situation of a split-estate, where a private entity holds rights to the surface property and the subsurface mineral estate is federally owned, the owner of the lease to the mineral estate is given priority to extract those resources.

<u>Problems with the Transdisciplinary Approach as a Solution:</u>

A transdisciplinary approach with respect to the current issue would necessitate representatives of all levels of decision makers, stakeholders, researchers, and society interacting and participating in “mutual learning” to resolve conflicts between these land use practices. As a skeptic of the Federal government’s ability to act efficiently, I do not see this option as being completely feasible as explained in Reyers et al. (2010). There are a number of laws and regulations that inhibit a <u>balanced</u> representation of values in land use planning. The BLM, as a federal organization, gets its marching orders from the Executive branch and Congress, which have been pushing domestic energy production. While this satisfies the energy production industry, specifically the production of fossil fuels, it may not represent appropriate action by way of environmentalist and ranching values.

Decentralization:

Nelson (1999) may find some of the same faults I have with this system of thought as presented and suggest decentralization of management of federal lands. Proponents of decentralization argue that it would allow for "a greater pluralism in management decisions, which can reflect the diversity of values held among Americans of many different backgrounds" (Nelson 1999). Federal management is too unwieldy when it comes to addressing the intricacies of values held by different interest groups or individuals in various regions. Our bipartisan system requires citizens to vote based on the <u>most</u> pressing issues to them. However when it comes to presidential candidates, for example, (who, under the right congressional circumstances, can greatly alter federal land management policies once elected [Forbis 2010]) that does not mean their policies align exactly with the voter's values. Under federal management, some people may need to "bow to the values of others" (Nelson 1999) who may or may not be in any way connected with local situations.

<u>Problems with Decentralization as a Solution:</u>

On the other hand, decentralization also presents a number of other problems. If power is given to state governments, the source and amount of funding will change and maybe not for the better as far as some interest groups are concerned. When faced with this option in both the 1970s and 1980s (Sagebrush Rebellion) and 1995, previous supporters of decentralization failed to follow through (Nelson 1999). Ranchers continued to support federal control with the fear that the costs of grazing on now state land would increase (Nelson 1999). Similarly, environmentalists opposed construction of dams, highways, and power plants proposed by authorities at the federal level in the 1960s but continue to rely on federal power to, for example, establish national parks and wildlife refuges (Nelson 1999). Interest groups of one kind or another seem to fear losing some benefits associated with federal control if local values are upheld.

As a compromise, Nelson (1999) discusses the possibility of still further solutions, including public lands corporations and giving more power to the local and regional field offices of existing federal agencies, such as the BLM. While these and the previously mentioned solutions have certain benefits, none of them are perfect and certainly would not be sufficient on their own. It will likely take a combination of policy changes as well as a shift in our values as U.S. citizens to ease the conflict between multiple land use practices on public lands.

<u>Conclusion:</u>

Ecological research, both basic and applied, can inform management decisions on public land in a number of ways. Most importantly, it can illuminate any negative effects of a given land use practice as well as the causes behind that effect. This type of information can be important to a management agency, such as the BLM, with a multi-use mission as these studies indicate under what management regimes a land use is in contradiction with other goals, such as conservation or restoration.

The current body of research, however, is flawed. In order to make fully informed decisions, land managers are in need of site or ecosystem-specific studies, which may not be available for the ecosystem in question. In addition, as is the case with investigations of the effects of extraction of natural gas, lack of baseline data and systematically controlled experiments lead to incomplete answering of questions pertinent to land managers. For example, when managing fracking practices, the BLM would be better served by studies examining the mechanism of contamination of air and water: is contamination an intrinsic result of the fracturing process or could the problem be rectified with further safety measures and maintenance of machinery?

To produce research that is more pertinent to land managers, researchers and managers can work together more closely. This could be facilitated if funding were available to BLM field offices to solicit investigation into questions <u>they</u> need answered locally. This may necessitate a certain level of decentralization or at least more discretionary power given to local managers within the agency. Close collaboration between researchers and land managers from the beginning would ensure the produced results could better inform management decisions.

Public land managers of the BLM cannot only consider scientific research when making land use decisions, however. Its multi-use mission statement requires an integration of conservation, restoration, recreation and resource use and extraction. This can lead to a number of conflicts or contradictions between goals. In addition, national, state, and local values and priorities play into which land use practices are deemed acceptable, often regardless of scientific research. In order to remedy the situation, boundary spanning, a transdisciplinary approach, and decentralization have been suggested. Though each of these approaches has their flaws, perhaps aspects of each could be combined to produce an acceptable solution. Though it is not likely to come easily, some fusion of policy, research, and value changes is needed to ease tensions regarding the multiple uses and management of public land.

Acknowledgements: I would like to thank all the people that helped make this thesis a reality, especially Dr. Frances Hanzawa for her prompt feedback, flexibility, and guidance. I would also like to thank Char Miller, the rest of the Environmental Analysis Department, and my friends and family who helped me stay motivated when I thought it would never be done.

Figures:

Fig. 1 Public Land in the U.S. Map depicts only surface estate managed by BLM and within other public land systems. Retrieved from the Native Forest Council's "Vision of Publically Owned Lands" (http://www.forestcouncil.org/learn/index.html)

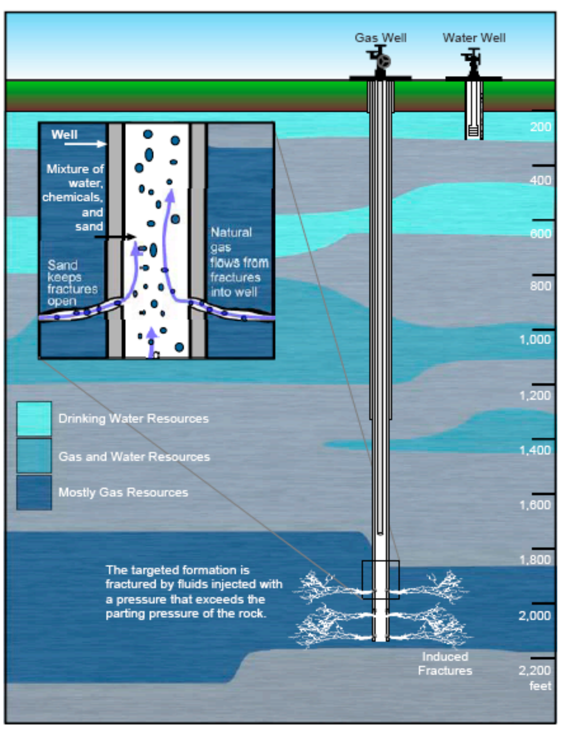

Fig. 2 Hydraulic fracturing injects large amounts of water, chemicals, and sand into wells to fracture shale. Retrieved from EPA 2011, Fig. 7

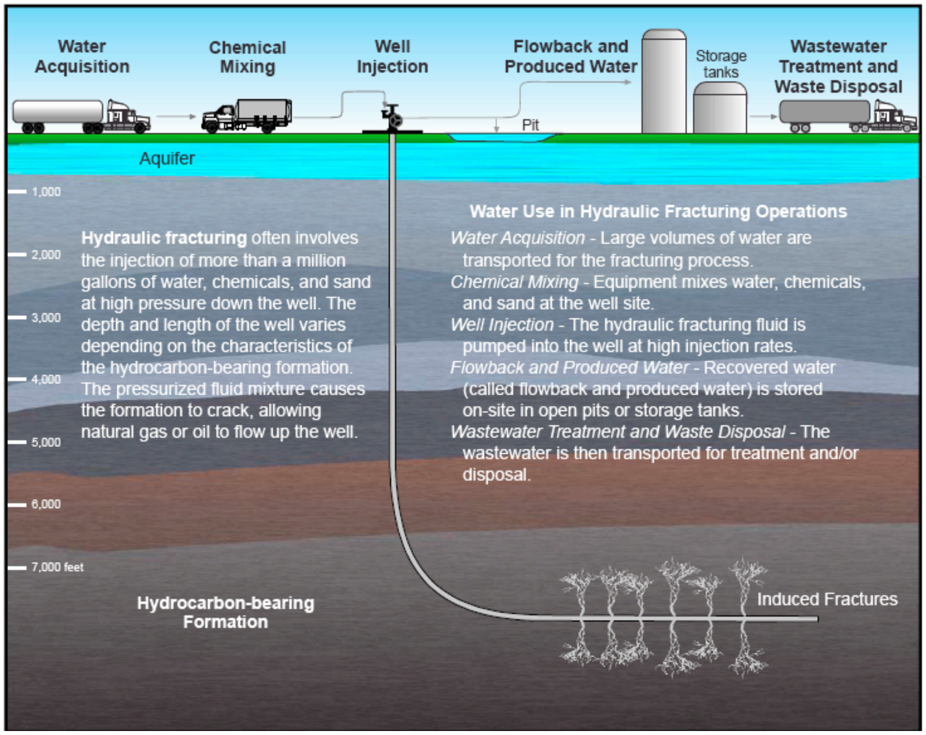

Fig. 3 Water Lifecycle in Hydraulic Fracturing. Retrieved from EPA 2011, Fig. 6

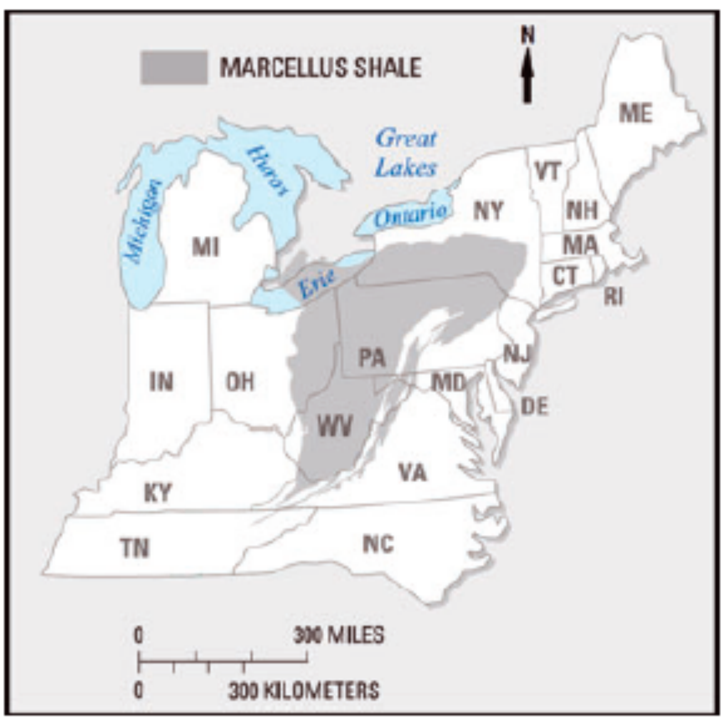

Fig. 4 Marcellus Shale Formation. Retrieved from Rozell and Reaven (2012) Fig. 1

Work Cited:

Allert, A. L., J. F. Fairchild, C. J. Schmitt, J. M. Besser, W. G. Brumbaugh, and S. J. Olson. 2009. "Effects of Mining-derived Metals on Riffle-dwelling Benthic Fishes in Southeast Missouri, USA." *Ecotoxicology and Environmental Safety* 72: 1642–1651. doi:10.1016/j.ecoenv.2009.02.014.

Arthur, J. D., B. Bohm, and M. Layne. 2008. "Hydraulic fracturing considerations for natural gas wells of the Marcellus shale." The Ground Water Protection Council 2008 Annual Forum, Cincinnati, OH, September 21-24, 2008, 16 pp

Bamberger, M., and R. E. Oswald. 2012. "Impacts of gas drilling on human and animal health." *New Solutions* 22: 51-77

Beck, J. L., and D. L. Mitchell. 2000. "Influences of Livestock Grazing on Sage Grouse Habitat." *Wildlife Society Bulletin* 28: 993–1002.

Bremer, D. J., J. M. Ham, C. E. Owensby, and A. K. Knapp. 1998. "Responses of soil respiration to clipping and grazing in a tallgrass prairie." *Journal of Environmental Quality*. 27: 1539–1548.

Bryner, G. C. 1998. *U.S. Land and Natural Resources Policy: a Public Issues Handbook*. Greenwood Press.

Carlson, L. W., and A. T. Cringan. 1975. "Oil Shale Development and Wildlife in Northwestern Colorado." *Wildlife Society Bulletin* 3: 7–12. doi:10.2307/3781709.

Clark, C. E., and J. A. Veil. 2009. "Produced water volumes and management practices in the United States." United States Department of Energy, Argonne National Laboratory. Web. www.evs.anl.gov/pub/doc/ANL_EVS__R09_produced_water_volume_report_2437.pdf

Coalition of Citizens Who Love Colorado (CCLC). "Wildlife Impacts." *Save Colorado From Fracking*. Web. http://www.savecoloradofromfracking.org/harm/wildlife.html

Collins, S. L., A. K. Knapp, J. M. Briggs, J. M. Blair, and E. M. Steinauer. 1998. "Modulation of Diversity by Grazing and Mowing in Native Tallgrass Prairie." *Science* 280: 745–747. doi:10.1126/science.280.5364.745.

Cypher, B. L., G. D. Warrick, M. R. M. Otten, T. P. O'Farrell, W. H. Berry, C. E. Harris, T. T. Kato, P. M. McCue, J. H. Scrivner, and B. W. Zoellick. 2000. "Population Dynamics of San Joaquin Kit Foxes at the Naval Petroleum Reserves in California." *Wildlife Monographs* (145): 1–43. doi:10.2307/3830818.

Dammel, J. A., J. M. Bielicki, M. F. Pollak, and E. J. Wilson. 2011. "A Tale of Two Technologies: Hydraulic Fracturing and Geologic Carbon Sequestration." *Environmental Science & Technology* 45: 5075–5076. doi:10.1021/es201403c.

Datangel, B., and J. L. Goldfarb. 2011. "Heavy Metals in Colorado and Chinese Oil Shale Semicoke: Disposal Issues, Impediments to Byproduct Conversion." *Energy & Fuels* 25: 3522–3529. doi:10.1021/ef2006386.

Debatin, B. 2012. "New Study: Severe health impact of fracking." *Slow Down Fracking in Athens County (SD-FRAC)*. Web. http://slowdownfracking.wordpress.com/2012/01/12/health-impact-of-fracking/

Driscoll, C. T., K. F. Lambert, F. S. Chapin III, D. J. Nowak, T. A. Spies, F. J. Swanson, D. B. Kittredge, and C. M. Hart. 2012. "Science and Society: The Role of Long-Term Studies in Environmental Stewardship." *BioScience* 62: 354–366. aph.

Evans, R. A., and J. A. Young. 1989. "Characterization and analysis of abiotic factors and vegetation." Grassland structure and function. Kluwer Academic Publishers, Boston. Pages 13-27 in L. F. Huenneke and H. A. Mooney, editors.

Fitzsimmons, A. K. 2012. *Reforming Federal Land Management: Cutting the Gordian Knot*. Rowman & Littlefield Publishers.

"FLPMA.pdf." 2012. Accessed: http://www.blm.gov/flpma/FLPMA.pdf.

Forbis, R. 2009. "Drill, baby, drill: An analysis of how energy development displaced ranching's dominance of the BLM's land-use subgovernment." *Western Political Science Association* (Conference Papers)

Forrest, R. 2011. "No more drilling in the dark: Exposing the hazards of natural gas production and protecting America's drinking water and wildlife habitats." *National Wildlife Federation* 2011 report.

Foster, B. L., and K. L. Gross. 1998. "Species Richness in a Successional Grassland: Effects of Nitrogen Enrichment and Plant Litter." *Ecology* 79: 2593–2602. doi:10.2307/176503.

Germano, D. J., G. B. Rathbun, and L. R. Saslaw. 2001. "Managing Exotic Grasses and Conserving Declining Species." *Wildlife Society Bulletin* 29: 551–559. doi:10.2307/3784180.

Goguen, C. B., and N. E. Mathews. 2001. "Brown-Headed Cowbird Behavior and Movements in Relation to Livestock Grazing." *Ecological Applications* 11: 1533–1544. doi:10.1890/1051-0761(2001)011[1533:BHCBAM]2.0.CO;2.

Gregory, K. B., R. D. Vidic, and D. A. Dzombak. 2011. "Water Management Challenges Associated with the Production of Shale Gas by Hydraulic Fracturing." *Elements* 7: 181–186. doi:10.2113/gselements.7.3.181.

Holland, E. A., and J. K. Detling. 1990. "Plant Response to Herbivory and Belowground Nitrogen Cycling." *Ecology* 71: 1040–1049. doi:10.2307/1937372.

The Investigative Newswire. 2012. "Colorado drilling projects and BLM placed under the lens." *Public Herald*. Web. http://www.publicherald.org/archives/16435/investigative-reports/energy-investigations/

Jackson, R. B., S. G. Osborn, A. Vengosh, and N. R. Warner. 2011. "Reply to Davies: Hydraulic Fracturing Remains a Possible Mechanism for Observed Methane Contamination of Drinking Water." *Proceedings of the National Academy of Sciences* 108: E872–E872. doi:10.1073/pnas.1113768108.

Jaramillo, P., W. M. Griffin, and H. S. Mathews. 2007. "Comparative life-cycle air emissions of coal, domestic natural gas, LNG, and SNG for electricity generation." *Environmental Science and Technology* 41: 6290-6296

Kauffman, J. B., W. C. Krueger, and M. Vavra. 1983. "Effects of Late Season Cattle Grazing on Riparian Plant Communities." *Journal of Range Management* 36: 685–691. doi:10.2307/3898185.

Kearns, C., and D. Oliveras. 2009. "Environmental Factors Affecting Bee Diversity in Urban and Remote Grassland Plots in Boulder, Colorado." *Journal of Insect Conservation* 13: 655–665. doi:10.1007/s10841-009-9215-4.

Kimball, S., and P. M. Schiffman. 2003. "Differing Effects of Cattle Grazing on Native and Alien Plants." *Conservation Biology* 17: 1681–1693. doi:10.2307/3588915.

Klebenow, D. A. 1985. "Habitat management of sage grouse in Nevada." *World Pheasant Association Journal* 10:34-46

Lauenroth, W. K. and D. G. Milchunas. 1991. "The shortgrass stepp. – In: Coupland, R. T. (ed/), Natural grasslands; introduction and western hemisphere." *Ecosystems of the World* 8A:183-226

LeCain, D. R, J. A. Morgan, G. E. Schuman, J. D. Reeder, and R. H. Hart. 2002. "Carbon Exchange and Species Composition of Grazed Pastures and Exclosures in the Shortgrass Steppe of Colorado." *Agriculture, Ecosystems & Environment* 93: 421–435. doi:10.1016/S0167-8809(01)00290-0.

Lengwiler, M. 2006. "Between charisma and heuristics: four styles of interdisciplinarity." *Science and Public Policy* 33: 423-434

Long Term Ecological Research Network (LTER). (2012). "Core research areas." Web. http://www.lternet.edu/research/core-areas

Max-Neef, M. A. 2005. "Foundations of transdisciplinarity." *Ecological Economics* 53: 5-16.

McKenzie, L. M., R. Z. Witter, L. S. Newman, and J. L. Adgate. 2012. "Human health risk assessment of air emissions from development of unconventional natural gas resources." *Science of the Total Environment* 424: 79–87. doi:10.1016/j.scitotenv.2012.02.018.

Milchunas, D. G., and W. K. Lauenroth. 1993. "Quantitative Effects of Grazing on Vegetation and Soils Over a Global Range of Environments." *Ecological Monographs* 63: 328–366. doi:10.2307/2937150.

Milchunas, D. G., W. K. Lauenroth, P. L. Chapman, and M. K. Kazempour. 1989. "Effects of Grazing, Topography, and Precipitation on the Structure of a Semiarid Grassland." *Vegetatio* 80: 11–23.

Milchunas, D. G., W. K. Lauenroth, and I. C. Burke. 1998. "Livestock grazing: Animal and plant biodiversity of shortgrass steppe and the relationship to ecosystem function." *Oikos.* 83: 65-74

Native Forest Council (NFC). 2010. "Native Forest Council's vision for publically owned lands." *Learn More*. Web. http://www.forestcouncil.org/learn/index.html

Nelson, R. H. 1999. "Public Lands: A System in Crisis." *Forum for Applied Research & Public Policy* 14: 64.

Orupõld, K., M. Heinlaan, L. Põllumaa, H.Dubourguier, and A. Kahru. 2012. "Impact of Oil Shale Opencast Mining and Combustion on Narva River and Its Tributaries: Chemical and Ecotoxicological Characterisation." *Oil Shale* 29: 173–189. buh.

Osborn, Stephen G., Avner Vengosh, Nathaniel R. Warner, and Robert B. Jackson. 2011a. "Methane Contamination of Drinking Water Accompanying Gas-well Drilling and Hydraulic Fracturing." *Proceedings of the National Academy of Sciences* 108: 8172–8176. doi:10.1073/pnas.1100682108.

Osborn, S. G, A. Vengosh, N. R. Warner, R. B. Jackson. 2011b. "Reply to Saba and Orzechowski and Schon: Methane Contamination of Drinking Water Accompanying Gas-well Drilling and Hydraulic Fracturing." *Proceedings of the National Academy of Sciences of the United States of America* 108: E665–E666. doi:10.1073/pnas.1109270108.

Painter, E. L., and J. K. Detling. 1981. "Effects of Defoliation on Net Photosynthesis and Regrowth of Western Wheatgrass." *Journal of Range Management* 34: 68–71. doi:10.2307/3898458.

Piñeiro, G., J. M. Paruelo, M. Oesterheld, and E. G. Jobbágy. 2010. "Pathways of Grazing Effects on Soil Organic Carbon and Nitrogen." *Rangeland Ecology & Management* 63: 109–119.

Reeder, J. D., G. E. Schuman, J.A. Morgan, and D. R. LeCain. 2004. "Response of Organic and Inorganic Carbon and Nitrogen to Long-Term Grazing of the Shortgrass Steppe." *Environmental Management* 33: 485–495. doi:10.1007/s00267-003-9106-5.

Reyers, B., D. J. Roux, R. M. Cowling, A. E. Ginsburg, J. L. Nel, and P. O Farrell. 2010. "Conservation Planning as a Transdisciplinary Process." *Conservation Biology* 24: 957–965. aph.

Rozell, D.l J., and S. J. Reaven. 2012. "Water Pollution Risk Associated with Natural Gas Extraction from the Marcellus Shale." *Risk Analysis* 32: 1382–1393. doi:10.1111/j.1539-6924.2011.01757.x.

Saba, T., and M. Orzechowski. 2011. "Lack of Data to Support a Relationship Between Methane Contamination of Drinking Water Wells and Hydraulic Fracturing." *Proceedings of the National Academy of Sciences* 108: E663–E663. doi:10.1073/pnas.1108435108.

Schon, S. C. 2011. "Hydraulic Fracturing Not Responsible for Methane Migration." *Proceedings of the National Academy of Sciences* 108: E664–E664. doi:10.1073/pnas.1107960108.

Skillen, J. 2009. *The Nation's Largest Landlord: The Bureau of Land Management in the American West*. Lawrence, Kan: University Press of Kansas.

Sørensen, L. I., J. Mikola, M. M. Kytöviita, and J. Olofsson. 2009. "Trampling and spatial heterogeneity explain decomposer abundances in a sub-Arctic grassland subjected to simulated reindeer grazing." *Ecosystems* 12: 830-842

Stevens, B. 2012. "The facts about fracking fluid and its disposal." *OilPrice*. Web. http://oilprice.com/Energy/Energy-General/The-Facts-about-Fracking-Fluid-and-its-Disposal.html

Teal, J. M. and R. W. Howarth. 1984. "Oil Spill Studies: A Review of Ecological Effects." *Environmental Management* 8: 27–43. doi:10.1007/BF01867871.

U.S. Bureau of Land Management (BLM). 1996. "Carrizo Plain National Area management plan." BLM, Bakersfield, California. Web. http://www.blm.gov/ca/st/en/fo/bakersfield/Programs/planning/cpnm_plan.html

U.S. Bureau of Land Management (BLM). 2008. "Get Energized." Colorado. Web. http://www.blm.gov/co/st/en/BLM_Programs/energy/get_energized.html

U.S. Bureau of Land Management (BLM). 2009. "Operations and production: Monitoring, inspection and enforcement." U.S. Department of the Interior. Web. http://www.blm.gov/wo/st/en/prog/energy/oil_and_gas/leasing_of_onshore/og_production.html

U.S. Bureau of Land Management (BLM). 2011a. "Fracking on BLM Colorado Well Sites." www.blm.gov/pgdata/etc/.../blm/co/...Par.../Fracking_March11.pdf

U.S. Bureau of Land Management (BLM). 2011b. "The BLM: A sound investment for America." www.blm.gov/or/.../files/BLM-SoundInvestmentforAmerica.pdf

U.S. Bureau of Land Management (BLM). 2012. "Fact sheet on the BLM's Management of Livestock grazing." *Grazing*. Web. http://www.blm.gov/wo/st/en/prog/grazing.html.

U.S. Energy Information Administration (EIA). 2011. "Annual energy outlook 2011." United States Department of Energy, Energy Information Administration DOE/EIA-0383(2011): www.eia.doe.gov/forecasts/aeo

U.S. Energy Information Administration (EIA). 2012. "What is shale gas and why is it important?" *Energy in Brief*. Web. http://www.eia.gov/energy_in_brief/article/about_shale_gas.cfm

U.S. Environmental Protection Agency (EPA). 2011. "Plan to study the potential impacts of hydraulic fracturing on drinking water resources." Office of Research and Development Washington, D.C. Web. http://www.epa.gov/hfstudy/

U.S. House of Representatives Committee on Energy and Commerce Minority Staff. 2011. "Chemicals used in hydraulic fracturing." democrats.energycommerce.house.gov/.../Hydraulic%20Fracturing%20Report%204.18.11.pdf

Vergano, D. 2012. "The EPA is clamping down on gas drilling smog." *Business Insider.* Web. http://articles.businessinsider.com/2012-04-19/news/31366135_1_fracking-natural-gas-initial-drilling

Weber, J. G. 2012. "The Effects of a Natural Gas Boom on Employment and Income in Colorado, Texas, and Wyoming." *Energy Economics* 34: 1580–1588.

<u>**Appendix I: Calhoun et al. (2012)**</u>

Does nutrient enrichment interact with Castilleja miniata to alter plant communities in montane meadows?

Student: Corinne Calhoun

Mentors: Aimée Classen and Nathan Sanders

Advanced Independent Research, NSF funded REU

Summer 2012

ABSTRACT

Many studies have demonstrated the negative effects of hemiparasites on host performance, such as decreased host biomass and reproductive capabilities. Much less work, however, has examined the role of hemiparasites in shaping plant community structure, though it has been proposed that the effect of a hemiparasite can vary based on a number of factors, including nutrient availability. Using *Castilleja miniata* as a test species, I tested for an interactive effect of hemiparasite removal and nitrogen enrichment on plant community structure. I found no evidence of an interactive effect altering species richness, total plant cover, diversity, or evenness. Nitrogen availability and *C. miniata* presence or absence may have an additive effect on nitrogen-fixer productivity. Nitrogen-fixers grew more in plots under ambient soil conditions where *C. miniata* was removed. The importance of nitrogen-fixers in an ecosystem warrants further research into the dynamics between nutrient cycling, hemiparasites, and nitrogen-fixer performance.

INTRODUCTION

There are over 3,000 species of flowering parasitic plants (Bardgett et al. 2006) and they inhabit almost all major terrestrial ecosystems (Adler 2002). A hemiparasite is a kind of parasitic plant that not only gains carbon, nutrients, water, and defensive compounds from host plants, but also can photosynthesize and gain nutrients non-parasitically (Press and Phoenix 2005; Adler 2003 citing Arslanian *et al.* 1990).

Many studies have investigated the effects of hemiparasites on host performance (e.g. Seel and Press 1996; Matthies 1995; Mudrák and Leps 2012) and root hemiparasites, such as witchweeds (*Striga spp.*), have been shown to reduce the yield of major cereal crops (Malcom 1966; Press 1998 citing Parker and Riches 1993). In their two-year lab study, Seel and Press (1996) found that grasses infected with the hemiparasite *Rhinanthus minor* had less than half the mass of non-infected grasses. Matthies (1995) also found negative effects of a hemiparasite on host growth rates.

However, despite the plethora of knowledge pertaining to the effects of a hemiparasite on a host, the mechanisms by which a host is affected and the community-level effects of a hemiparasite are not as well understood. By parasitizing a host plant, a hemiparasite can decrease its host's biomass (Mudrák and Leps 2010) and ability to compete with neighboring non-host species, thus changing community dynamics (Bardgett et al. 2006). If the hemiparasite negatively affects the more competitively dominant or common species in the community, this may allow non-host species to increase in abundance or biomass and positively affect the plant community diversity (Mudrák and Leps 2010).

There is evidence, however, that hemiparasite influence on plant community diversity may change with habitat type and resource availability (Gibson and Watkinson 1992; Press 1998; Westbury et al 2006). Gibson and Watkinson (1991) suggested that host selectivity in *R. minor* may depend on a number of factors, including the relative abundance of hosts, environmental conditions, and genetic structure of both host and parasite populations. Under laboratory conditions, they found decreased root connectivity between *R. minor* and its host when grown in nitrogen (N) rich soil as well as greater host biomass relative to hosts grown in N poor soil with *R. minor*. This variation in root connectivity suggests that hemiparasite colonization as well as the impact its colonization has on the surrounding plant community can vary based on nutrient availability. For example, at higher nutrient levels both host and hemiparasite invest less in their root system: the hemiparasite advantage of gaining nutrients from a host plant decreases (Mudrák and Leps 2010), possibly along with the negative effects on the host plant and cascading community level effects.

This study will further investigate how a hemiparasite structures plant communities under altered nutrient conditions by manipulating soil N availability and presence or absence of the long-lived, subalpine perennial root hemiparasite *Castilleja miniata* (Adler 2003). Nitrogen availability is cited as one of the most common nutrients that limit plant growth (Bowman *et al*. 1993). Manipulation studies in non-hemiparasite communities have shown an increase in total aboveground biomass with the addition of N and a decrease in total aboveground biomass with reduced N availability (Blue et al. 2011 citing Craine et al. 2003; Gruner et al. 2008; Cleland and Harpole 2010; Wedin and Tilman 1993; and Throop 2005). Chalcraft et al. (2008) demonstrated a scale-dependent effect of N enrichment on biodiversity: on a small scale, an increase in primary productivity due to N enrichment leads to a decrease in diversity.

While *Castilleja* spp. can parasitize a wide range of hosts (Spasojevic and Suding 2011), factors contributing to host selectivity and the community-level effects of the hemiparasite are still not well understood. It has been demonstrated that *Castilleja miniata* can increase community evenness: previous work in this ecosystem found that removal of *Castilleja spp.* decreased evenness of the surrounding plant community but had no significant effect on species richness (Reed et al., unpublished data). If increasing available nutrients shifts the relationship between *Castilleja* and its hosts, as it does in laboratory studies with *R. minor*, I predict an observable change in *Castilleja's* effect on the surrounding plant community, specifically community evenness.

In this study, I manipulated the direct and interactive effects of nutrient availability and *Castilleja* presence on total plant cover, plant community diversity, and evenness. Suding et al (2006) demonstrated the importance of species function in removal manipulations and the community's response to species loss, so I also broke down treatment effects based on functional groups (non N-fixer forbs, graminoids, and N-fixers) that, due to their functional differences, may respond to N enrichment and *Castilleja* removal differently. I hypothesized that (i) *Castilleja* removal and N addition will have an interactive effect and increase total plant percent cover and decrease evenness of the plant community ii) N addition will have a main effect of increasing overall percent plant cover, decrease percent cover of N-fixers, and decrease the evenness of the plant community. iii) *Castilleja* removal will increase the percent cover of one or a few species, likely within the same functional group, and decrease evenness of the plant community. Unlike many previous studies, this manipulation illuminates some of the factors determining the effect of a hemiparasite on a plant community in a non-laboratory setting.

METHODS

Study site: We established this study in an aspen meadow near the Rocky Mountain Biological Laboratory in Gothic, Colorado, USA. This site is roughly 3000 meters above sea level and is located along the Judd Falls trail, approximately 200 meters from the trailhead. Soils are rocky and well drained. I established experimental plots in three distinct sites located within 25 meters of one another. Sites were chosen based on the location of abundant patches of Castilleja miniata and all sites have roughly SW facing slopes. Across all sites, the most abundant species were Castilleja miniata, *Carex spp.*, Erigeron speciosus, and Potentilla pulcherrima.

Experimental design: Each of the 24 experimental plots was 1.5 ×1.5 m with a minimum 1 m buffer area between plots. I randomly assigned plots to one of four treatments: (1) *Castilleja* removal, (2) N addition, (3) Castilleja removal × N addition, and (4) control plots. Prior to treatment establishment, we measured percent cover of each species in a 1 × 1 m quadrat within each 1.5 × 1.5 m plot, in addition to the cover of Castilleja within each plot. Ten days and twenty-three days after I applied the treatments, we measured percent cover of every species within each plot.

Castilleja removal: I removed all aboveground Castilleja biomass using hand-held pruning shears from each 1.5 m^2 plot assigned the removal treatment. If Castilleja sulphurea was present, it was also removed. I regularly clipped Castilleja removal plots throughout the summer.

Nitrogen addition: On the same day as the original Castilleja removal, I added N to the assigned 1.5m^2 N-addition plots using 30-0-3 (30% N, 0% phosphorus, 3% potassium) lawn fertilizer. Each N-addition plot was sprinkled with fertilizer once during the summer at a density of 20 g urea N per m^2 as per the methods in Blue et al. (2011).

Soil measurements: In addition to percent cover, I also measured soil moisture (0-12 cm) as percent volumetric water content (%VWC) and temperature (0-10 cm, ˚C) to ensure plots did not vary in this regard. On any given day, neither temperature nor %VWC varied significantly by treatment (Table 1). I measured soil moisture using a hand-held probe and temperature in ^{o}C with an analog thermometer. I averaged two soil moisture measurements taken at random locations in each plot and measured temperature from the center of each plot, both at a depth of 7 cm due to rockiness of soil. I took these measurements jointly three times throughout the study.

Calculations: From the percent cover measurements, I calculated 1) species richness 2) the Shannon-Wiener diversity index ($H' = \sum p_i \ln p_i$), and 3) evenness using the equation: Evenness=$(1/\Sigma p_i^2)(1/S)$ where p_i is the proportion of each species and S is the total number of species. All calculations were done in the statistical program R. I classified each species found in the plots as a non-N-fixer forb, graminoid (families Poaceae and Cyperaceae), or N-fixer (family Fabaceae) then calculated the Mean Cover Ratio within each treatment type of 1) total percent cover and 2) that of each functional group using the equation: Mean Cover Ratio=$(\Sigma(p_{2t}/p_{1t}))/P$ where p_{1t} is the total percent cover of all concerned species in plot X pre-treatment application, p_{2t} is the percent cover of all concerned species in plot X twenty-three days after treatment application, and P is the number of plots within the treatment type.

Statistical analyses: I ran four separate ANOVAs with the statistical program JMP 10 to analyze the effect of treatment type on species richness, diversity, evenness and total plant percent cover. I also analyzed the effect of treatment on Mean Cover Ratio of functional groups and total plant cover with four ANOVAs.

RESULTS

What effect do Castilleja *removal and N-addition have on total plant percent cover, species richness, diversity, and evenness of the entire plant community?*

Ten days after treatment application, I found no significant difference in species richness, total plant percent cover, diversity, or evenness of the plant community based on *Castilleja* and N treatment (Table 2). Similarly, *Castilleja* and N treatment had no interactive or main effects on species richness, diversity, or evenness twenty-three days after treatment application (Table 2, Fig. 1a-1d). In other words, plots did not vary in total plant percent cover, diversity, or evenness based on treatment type at any time after treatment application. All following results are based on percent cover measurements taken twenty-three days after treatment application.

How does Castilleja *removal and N-addition affect total growth and growth of different functional groups in the community?*

Neither *Castilleja* removal nor N addition caused a different response in Mean Cover Ratio for total percent cover, percent cover of non N-fixer forbs, or percent cover of graminoids based on treatment (Fig. 2a-2c). In the case of graminoids and N-fixers, figures suggest that N addition had little or a negative effect on percent cover (Fig. 2b and 2c). There was, however, a difference in the Mean Cover Ratio of N-fixers ($F=4.0022_{3,23}$; $p=0.0220$): Growth of N-fixers was significantly less in N treated plots with *Castilleja* present versus in plots under ambient soil conditions where *Castilleja* was removed (Fig. 2d). In all cases, percent cover seemed to increase from pre-treatment to post-treatment measurements most in plots where *Castilleja* was removed and N was not added (Fig. 2a-2d).

DISCUSSION

Hemiparasites have been shown to increase the evenness of the surrounding plant community (Reed et al. 2011, unpublished data). However, the factors contributing to such community level effects are largely unstudied. Gibson and Watkinson (1992) suggested the effect a hemiparasite has on the plant community may be partially determined by nutrient availability. I therefore predicted that nutrient enrichment would alter the way in which the hemiparasite *Castilleja miniata* affects its plant community using percent cover, species richness, diversity, and evenness as response variables.

What effect does Castilleja *removal and N-addition have on total plant percent cover, species richness, diversity, and evenness of the entire plant community?*

I did not see an effect of treatment type on species richness, which was as expected based on the previous study done in this system (Reed et al. 2011, unpublished data). In contrast to Reed et al. (2011, unpublished), however, I did not see a decrease in evenness with the removal of *Castilleja*. This may be due to the time constraints of the present study. Seel and Press (1996) suggest the effects of a hemiparasite on its host endure even after hemiparasite death, so long-term results may yield stronger trends.

How does Castilleja *removal and N-addition affect total growth and growth of different functional groups in the community?*

Functional group analysis did not indicate the presence of an interaction effect between *Castilleja* treatment and N treatment as I had previously hypothesized. It did reveal, however, an additive effect of both the treatments on the mean N-fixer cover ratio. N-fixers grew less well in N enriched plots and increased more readily in plots where *Castilleja* was removed. The decreased growth found in N enriched plots is to be expected, as N-fixers only hold a competitive advantage in N poor ecosystems (Skogen et al. 2011). However, the main effects of N enrichment on graminoids, forbs and total plant cover, suggesting a small or negative growth response, were not consistent with the hypothesis that in an N-poor ecosystem, N enrichment would increase productivity and decrease diversity. It may be the case, then, that N was not the most limiting resource to plant growth in this system during the summer of 2012. At the time of this study, there was a particularly short winter preceding a dry summer. Instead of N, water may have been limiting plant growth this year.

Results also indicate a trend towards all functional groups growing more in plots under ambient soil conditions where *Castilleja* was removed. On average, *Castilleja* negatively affected all functional groups such that in its absence, these groups increased in cover. If *Castilleja* presence only decreases its hosts' biomass, this may be an indication that in this system *Castilleja* is demonstrating generalist host preference, instead of targeting a single group or species.

This study has demonstrated that when *Castilleja miniata* is present in this system, N-fixer performance decreases. N-fixers are keystone species that, unlike other functional groups, can bring in new N to the system. This has profound implications for hemiparasite systems if parasitization also decreases N-fixers' ability to perform this ecosystem function. Hemiparasites, however, have been shown to contain higher nutrient concentrations in their leaf litter and may serve the ecosystem by redistributing these nutrients: either by transferring them from comparatively N-rich hosts to other members of the community or unlocking slowly released nutrients from slow growing, long-lived perennial hosts (Press et al 1998). The effects of a hemiparasite on N-fixers and the overall plant community may depend heavily on host preference; therefore further research should investigate mechanisms of host preference and trace the transfer of nutrients between host, hemiparasite, and the community.

LITERATURE CITED

Arslanian, R. L., G. H. Harris, and F. R. Stermitz. New quinolizidine alkaloids from *Lupinus argenteus* and its hosted root parasite *Castilleja sulphurea*: stereochemistry and conformation of some naturally occurring cyclic carbinolamides. Journal of Organic Chemistry. 55. (1990): 1204-1210

Adler, L. S.. Host effects on herbivory and pollination in a hemiparasitic plant. Ecology. 83.10 (2002): 2700-2710

Adler, L. S.. Host species affects herbivory, pollination, and reproduction in experiments with parasitic Castilleja. Ecology. 84. (2003): 2083-2091

Bardgett, R. D., R. S. Smith, R. S. Shiel, S. Peacock, J. M. Simkin, H. Quirk and P. J. Hobbs. Parasitic plants indirectly regulate below-ground properties in grassland ecosystems. Nature. 439. (2006): 969-972

Bowman, W. D., T. A. Theodose, J. C. Schardt, R. T. Conant. Constraints of nutrient availability on primary production in two alpine tundra communities. Ecology 74.7 (1993): 2085-2097

Chalcraft, D. R., S. B. Cox, C. Clark, E. E. Cleland, K. N. Suding, E. Weiher, D. Pennington. Scale-dependent responses of plant biodiversity to nitrogen enrichment. Ecology 89.8 (2008): 2165-2171

Cleland, E. E., W. S. Harpole. Nitrogen enrichment and plant communities. Ann NY Academy of Sciences (2010): 46-61

Craine, J. M., P. B. Reich, D. Tilman, D. Ellesworth, J. Fargione, J. Knops, S. Naeem. The role of plant species in biomass production and response to elevated CO_2 and N. Ecology Letters 6. (2003): 623-630

Gibson, C. C. and A. R. Watkinson. Host selectivity and the mediation of competition by the root hemiparasite Rhinanthus minor. Oecologia. 86. (1991): 81-87

Gibson, C. C. and A. R. Watkinson. The role of the hemiparasitic annual Rhinanthus Minor in determining grassland community structure. Oecologia. 89.1 (1992): 62-68

Gruner, D. S., J. E. Smith, E. W. Seabloom, S. A. Sandin, J. T. Ngai, H. Hillebrand, W. S. Harpole, J. J. Elser, E. E. Cleland, M. E. S. Bracken, E. T. Borer, B. M. Bolker. A cross-system synthesis of consumer and nutrient resource control on producer biomass. Ecology Letters 11. (2008): 740-755

Malcom, W.M. Root parasitism of *Castilleja coccinea*. Ecology 47.2 (1966): 180-186

Matthies, D.. Parasitic and competitive interactions between the hemiparasites Rhinanthus serotinus and Odontites rubra and their host Medicago sativa. Ecology 83. (1995): 245-251

Mudrák, O., J. Leps. Interactions of the hemiparasitic species Rhinanthus minor with its host plant community at two nutrient levels. Folia Geobot 45. (2010): 407-424

Press, M. C.. Dracula or Robin Hood? A functional role for root hemiparasites in nutrient poor ecosystems. Oikos. 82.3 (1998): 609-611

Press, M. C. and G. K. Phoenix. Impacts of parasitic plants on natural communities. New Pathologist. 166. (2005): 737-751

Seel, W. E. and M. C. Press. Effects of repeated parasitism by Rhinanthus minor on the growth and photosynthesis of a perennial grass, Poa alpina. New Phytologist. 134. (1996): 495-502.

Skogen, K. A., K. E. Holsinger, Z. G. Cardon. Nitrogen deposition, competition and the decline of a regionally threatened legume, *Desmodium cuspidatum*. Oecologia. 165. (2011): 261-269

Spasojevic, M. J., and K. N. Suding. Contrasting effects of hemiparasites on ecosystem processes: can positive litter effects offset the negative effects of parasitism?. Oecologia. 165. (2011): 193-200

Suding, K. N, A. E. Miller, H. Bechtold, W. D. Bowman. The consequence of species loss on ecosystem nitrogen cycling depends on community compensation. Oecologia 149.1 (2006): 141-149

Throop, H. L.. Nitrogen deposition and herbivory affect biomass production and allocation in an annual plant. Oikos 111. (2005): 91-100

Wedin, D., D. Tilman. Competition among grasses along nitrogen gradient: initial conditions and mechanisms of competition. Ecological Monographs 63.2 (1993): 199-229

Westbury, D. B., A. Davies, B. A. Woodcock, N. P. Dunnett. Seeds of change: the value of using *Rhinanthus minor* in grassland restoration. Journal of Vegetation Science 17.4 (2006): 435-446

TABLES & FIGURES:

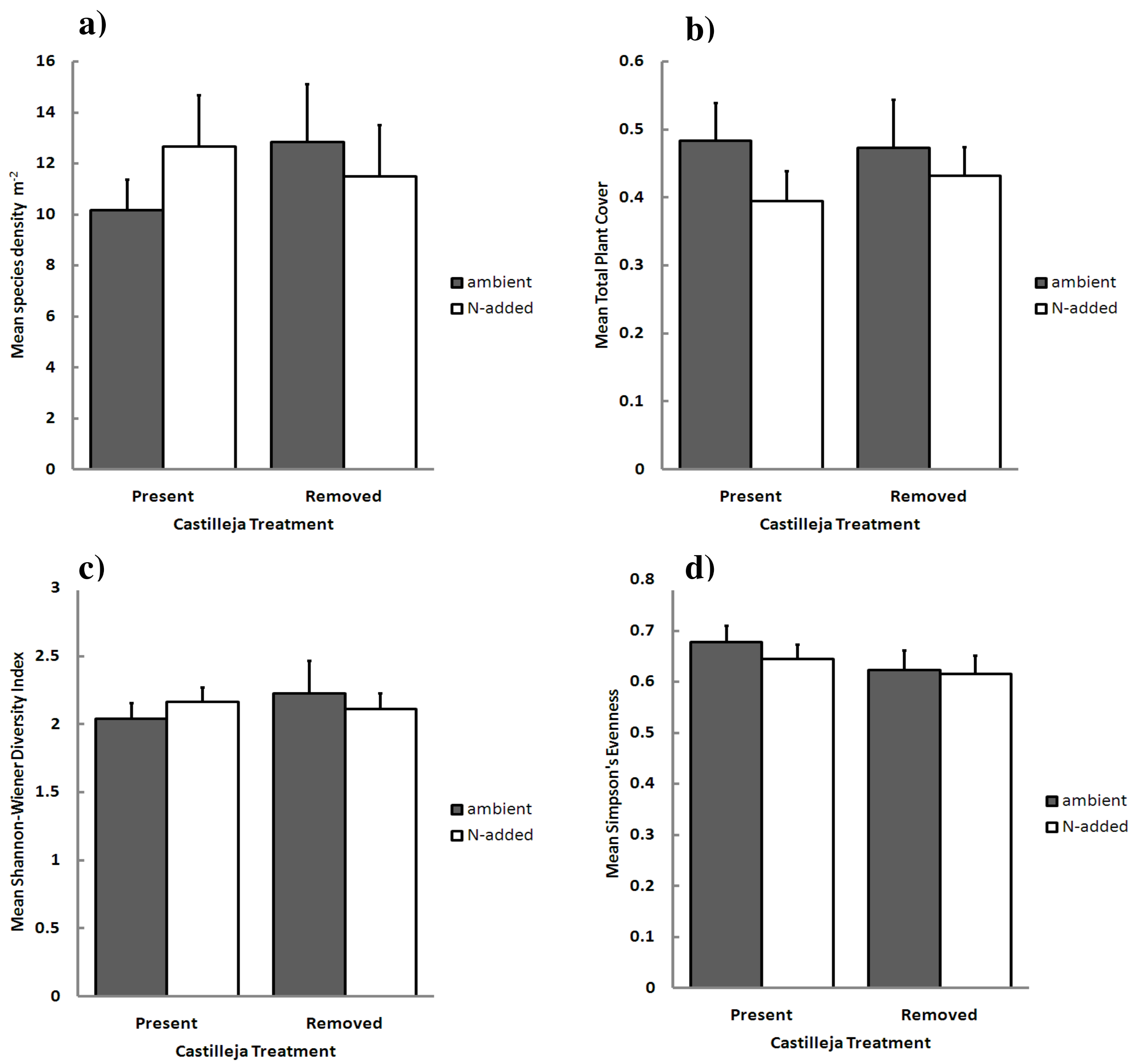

Figure 1. Effect of treatment type 23 days after treatment application on a) species richness ($F=0.5972_{3,23}$ $p=0.6242$), b) Mean total plant % cover ($F=0.5672_{3,23}$ $p=0.2541$), c) Mean Shannon-Wiener Diversity Index ($F=0.2617_{3,23}$ $p=0.9269$), and d) Mean Evenness ($F=0.6976_{3,23}$ $p=0.9410$), calculated as $(1/\Sigma p_i^2)(1/S)$ where p_i is the proportion of each species and S is the total number of species

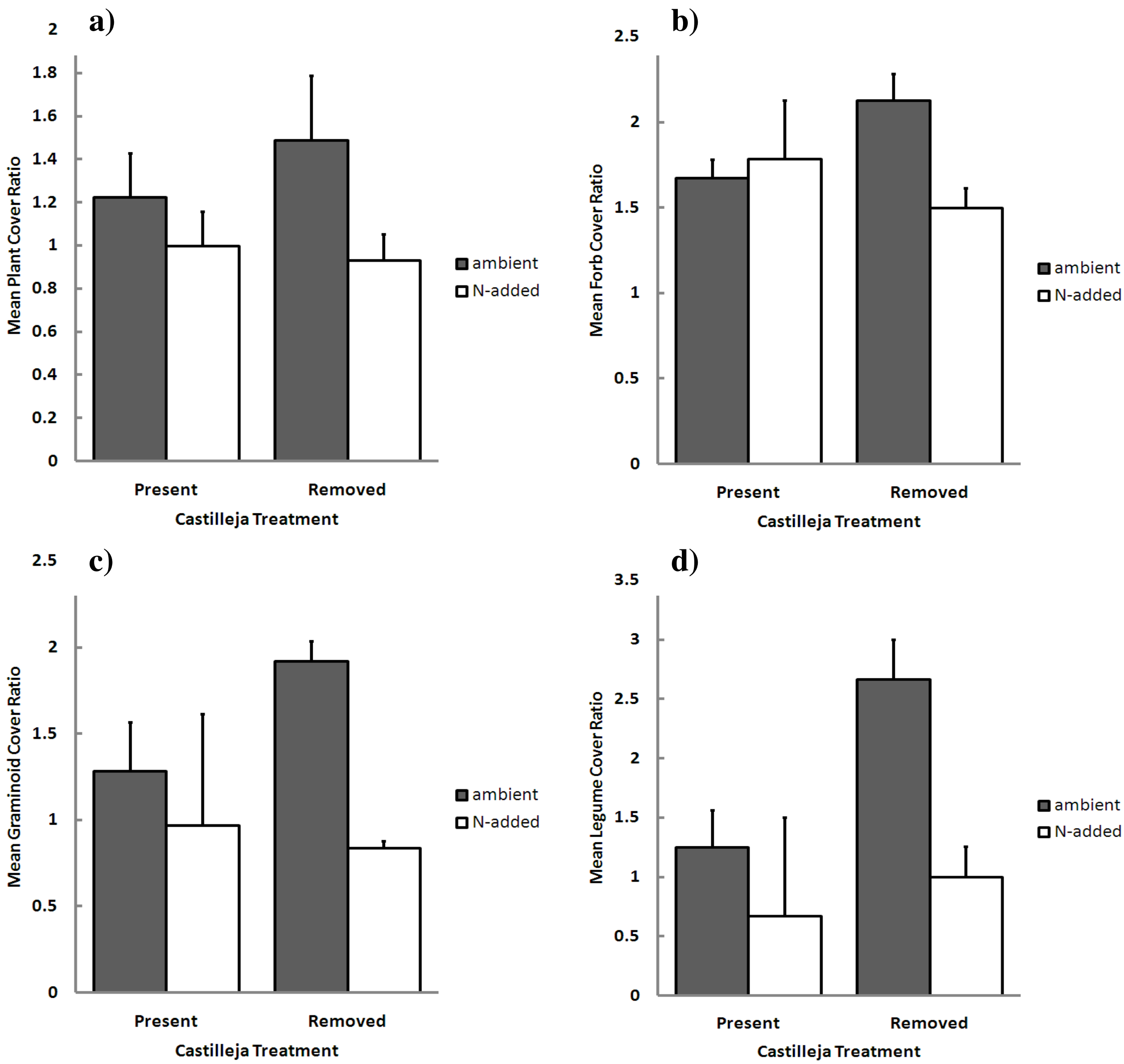

Figure 2. Effect of treatment type 23 days after treatment application where a value greater than 1 indicates growth over time, 1 indicates no growth, and a value less than 1 indicates negative growth. a) mean total plant cover ratio ($F=1.4651_{2,23}$ $p=0.2541$), b) mean non N-fixer forb cover ratio ($F=1.6658_{3,23}$ $p=0.2063$), c) mean graminoid cover ratio ($F=1.8118_{3,23}$ $p=0.1775$), and d) mean N-fixer cover ratio ($F=4.0022_{3,23}$ $p=0.0220$)

Table 1. Results of %VWC and Soil Temperature ANOVAs

DATE	[a]SOIL MOISTURE (%VWC at 7cm)	[b]SOIL TEMP (^{0}C at 7cm)
7/9/12	$F=0.7914_{3,23}$ $p=0.5129$	$F=1.3135_{3,23}$ $p=0.2977$
7/13/12	$F=1.1345_{3,23}$ $p=0.3591$	$F=0.8743_{3,23}$ $p=0.4709$
7/17/12	$F=0.5010_{3,23}$ $p=0.6859$	$F=0.7059_{3,23}$ $p=0.5597$

[a]On any given day, plot soil moisture measured as percent volumetric water content at 7cm did not vary by treatment
[b]On any given day, plot soil temperature measured in degrees Celsius at 7cm did not vary by treatment

Table 2. Results of Species Richness, Diversity, and Evenness ANOVAs

MEASUREMENT	TOTAL PLANT % COVER	[c] SPECIES RICHNESS	[d] DIVERSITY	[e] EVENNESS
Pre-treatment	$F=1.5846_{3,23}$ $p=0.2244$	$F=0.5078_{3,23}$ $p=0.6813$	$F=0.1525_{2,23}$ $p=0.9269$	$F=0.1302_{3,23}$ $p=0.9410$
[a]Post-treatment 1	$F=0.2775_{3,23}$ $p=0.8409$	$F=0.2978_{3.23}$ $p=0.8265$	$F=0.0931_{3,23}$ $p=0.9630$	$F=0.7128_{3,23}$ $p=0.5558$
[b]Post-treatment 2	$F=0.5672_{3,23}$ $p=0.6430$	$F=0.5972_{3,23}$ $p=0.6242$	$F=0.2617_{3,23}$ $p=0.7394$	$F=0.6976_{3,23}$ $p=0.5644$

[a] Post-treatment 1 indicates percent cover measurement taken 10 days after treatment application
[b] Post-treatment 2 indicates percent cover measurement taken 23 days after treatment application
[c] Total number of species found in plot does not vary by treatment before treatment application nor 10 and 23 days after
[d] Shannon-Wiener Diversity (diversity=) does not vary by treatment before treatment application nor 10 and 23 days after
[e] Evenness (simpson's evenness=$(1/\Sigma p_i^2)(1/S)$) does not vary by treatment before treatment application nor 10 and 23 days after